Chapter 5
Changes in the ecosystem services provided by forests and their economic valuation: a review

Cristina Marta-Pedroso, Lia Laporta, Vânia Proença, João C. Azevedo, and Tiago Domingos

Abstract In this chapter we discuss the trends in forest change and the associated drivers, the economic value of forests, the principles and challenges in evaluating the economic value of forests, and the role of valuation in informing decision-making. We address current major forest conservation initiatives at different scales and the mechanisms involved, whether supported by economic valuation or not. Today, 30 % of the world's forests are designated for productive functions, 24 % for multiple uses, 11.5 % for biodiversity conservation, 8.2 % for protective functions, and 3.7 % for social functions. The remaining 22.6 % are designated for other uses or remain unclassified. Global trends indicate that although the area of intensively managed forest continues to expand, the global extent of conservation and protective forests is also increasing as a result of political efforts to preserve and restore the ecological functions of forests. Forest management practices are potentially better supported by extended cost–benefit analyses that require an economic valuation of the whole array of benefits, whether market or non-market, provided by forests. Although we acknowledge other values and decision-making and support tools, the focus of the chapter is on the economic valuation approach. Our review in this chapter

C. Marta-Pedroso (✉) • J.C. Azevedo
CIMO—Centro de Investigação de Montanha, Escola Superior Agrária, Instituto Politécnico de Bragança, Campus de Santa Apolónia, Apartado 1172, 5301-855 Bragança, Portugal
e-mail: cmartapedroso@gmail.com

L. Laporta
IN+, Center for Innovation, Technology and Policy Research, Environment and Energy Section, DEM, Instituto Superior Técnico, Universidade Técnica de Lisboa, Avenida Rovisco Pais, 1, 1049-001 Lisbon, Portugal

CIBIO, Centro de Investigação em Biodiversidade e Recursos Genéticos, Universidade do Porto, Campus Agrário de Vairão, 4485-601 Vairão, Portugal

V. Proença • T. Domingos
IN+, Center for Innovation, Technology and Policy Research, Environment and Energy Section, DEM, Instituto Superior Técnico, Universidade Técnica de Lisboa, Avenida Rovisco Pais, 1, 1049-001 Lisbon, Portugal

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was guided by the goal of updating previous reviews of these topics. We have provided additional evidence that forests contribute to human well-being in many ways, and use the concept of ecosystem services as a building block to better understand, frame, and assess the economic benefits we derive from well-functioning forests.

5.1 Forest ecosystem services

Ecosystem services are broadly defined as the benefits that people obtain, directly or indirectly, from ecosystems (MEA 2005), and that contribute to human well-being when combined with other factors such as education, health care, and social equity.

The interdependency between ecological and social systems can be seen as a feedback loop: human well-being depends on the delivery of ecosystem services, but the capacity of ecosystems to deliver services depends on ecosystem conditions, which in turn are affected by society's choices about how to use ecosystem services and manage the ecosystems that provide them. These choices are greatly influenced by the level of human well-being and by the way society perceives and values ecosystem services.

Forests provide many ecosystem services, including supporting a large percentage of the world's biodiversity and contributing to human well-being at local (e.g., wood production), regional (e.g., groundwater recharge), and global (e.g., climate regulation) scales. The most easily understood and most quantifiable source of benefits derived from well-functioning forests pertains to the provision of goods and materials, even if their provision is not directly observed. Water, a basic and valuable good required for human existence, is a suitable first example. Though it is not obvious to the untrained eye, forests interact closely with and affect the hydrological cycle through evapotranspiration and their ability to increase infiltration into the soil by decreasing runoff; thus, forests are a key source of freshwater resources (Wang and Fu 2013). For example, about 80 % of the freshwater resources in the United States at the turn of the century originated from forests, which covered, at that time, about one-third of the country's surface area (USDA 1999). Human use and management of forest ecosystems can change the level of ecosystem services delivery and induce the production of one service to the detriment of others. This is the case for productive forests, which are planted and managed to produce timber, and the case for protective forests, which are planted or managed to prevent or reduce soil erosion.

Less quantifiable benefits that are often inadequately addressed include the benefits people obtain from forests through abstract concepts such as *esthetic*, *spiritual*, and *inspirational* values—which are called cultural services. Unlike timber production or soil erosion control, these benefits are not physically measurable. Instead, they take the form of *experiences* people can obtain from forests (Kareiva et al. 2011). Because they are intangible, communicating this category of benefits is more difficult, even when attempts are made to express the values in monetary terms. For forest ecosystems, a significant part of these benefits relates to recreational opportunities.

In southern Africa, for instance, trees play a crucial role in the cultural and spiritual lives of local communities (Sileshi et al. 2007), despite any hypothetical benefit they provide as tourist attractions. The inherent complexity of valuing people's experiences is well acknowledged in the literature (Boyd and Banzhaf 2007). We examine the methods for valuing ecosystem services more closely in Sect. 4.

5.2 Classification of ecosystem services and conceptual approaches

Exhaustively listing the whole array of benefits people obtain from ecosystems can be a challenging task, and some sort of labeling and operationalization of the concept was required at the beginning of efforts to conceptualize ecosystem services. Multiple classification systems for ecosystem services have evolved, and this variety has been justified by the premise that classification systems should focus on the purpose and context of the study (Costanza 2008). Notwithstanding, one of the most generally used classifications was described by the Millennium Ecosystem Assessment (MEA 2005), which distinguishes among four categories of services, which MEA defines as "the benefits that people derive from ecosystems": provisioning services (products obtained from ecosystems), regulating services (benefits obtained from the self-regulation of ecosystems), cultural services (non-material benefits obtained from ecosystems), and supporting services (services that are necessary for the production of other ecosystem services). Moreover, in this classification, biodiversity is understood as not only underpinning the ecosystem services but also as an ecosystem service itself; for example, medicinal plants are a provisioning service, whereas bird-watching is a cultural service.

Although this classification is still in use and it is generally accepted, it has some drawbacks. In particular, the consideration of supporting services as a separate category often leads to overlapping estimates and double-counting; this issue is a particular concern if economic valuation is to be undertaken, as we discuss later in the chapter. Another ecosystem service classification emerged from a more recent global initiative, *The Economics of Ecosystems and Biodiversity* (TEEB 2010). Although TEEB is similar to the MEA classification, TEEB considers the supporting services only as ecological processes, and introduces a new category called "habitat" services (TEEB 2010) to highlight the importance of ecosystems in providing habitat for migratory species and as gene-pool protectors. The TEEB classification and its approach differ from the MEA approach because TEEB explicitly aims to incorporate an economic analysis of changes in ecosystem services.

The Common International Classification of Ecosystem Services (Haines-Young and Potschin 2010) does not aim to replace existing classifications, but rather provides a framework that enables translation among different classifications and links to other classification systems that are used in economic and environmental accounting. Each classification has its own purposes, drawbacks, and advantages. The MEA and TEEB approaches are directed more at assessment and valuation of

ecosystem services, whereas the CICES approach was conceived as a system compatible with the design of integrated environmental and economic accounting methods (Maes et al. 2013). Those who advocate the use of CICES have pointed out that this classification at least potentially helps to overcome the problem of double-counting. This topic has been widely addressed in the literature (e.g., Boyd and Banzhaf 2007, Fisher et al. 2009, Mace and Bateman 2011), with debate focusing on the need to distinguish between services and benefits when economic valuation or environmental accounting is the purpose of the study. In essence, and regardless of the specific nomenclature adopted by each author in explaining their rationale, valuation should only be applied to things that are directly consumed by beneficiaries given that the values of ecological processes are already embedded in that final output. Some argue (Bateman et al. 2010) that if other input capitals are used to generate a benefit, they should be subtracted from the estimated value of the benefit to provide the net benefit.

Perhaps more important than finding a sovereign and unifying classification system or approach, we argue in this chapter that a deep understanding of the ecological dynamics of forest ecosystems is necessary to generate powerful insights into details of the chain of benefit delivery, and can therefore help managers to identify the best management options based on a more fully informed economic valuation.

5.3 Global trends and drivers of forest ecosystem services

5.3.1 Past, current, and future trends for forest systems

Human activity has caused the loss of about 40 % of the planet's original forests since preagricultural times, starting ca. 8000 years ago (Shvidenko et al. 2005). Temperate regions, such as Europe and North America, were particularly affected, losing more than 50 % of their natural forest cover before the mid-twentieth century (Fig. 5.1; Kaplan et al. 2009, MEA 2005). In tropical regions, the loss of forest cover has been less severe, but became a pervasive trend during the last half-century and will probably continue during the twenty-first century (Fig. 5.1; FAO 2012, MEA 2005).

In temperate and boreal regions, laws and policies to protect forests and to reverse deforestation emerged as a response to the shortage of timber and fuelwood and to the degradation of the forest's protective functions (Farrell et al. 2000, Rudel et al. 2005). Moreover, rural abandonment due to economic growth, improvements in agricultural efficiency, and the replacement of wood by fossil fuels as a source of energy also decreased pressures on forests (FAO 2012, Kaplan et al. 2009). Reforestation initiatives in the twentieth century, but also a few centuries ago, and natural forest regeneration following land abandonment restored much of the forest cover and helped to halt forest decline in temperate and boreal regions (Hobbs and Cramer 2007, Keenleyside et al. 2010, Rudel et al. 2005).

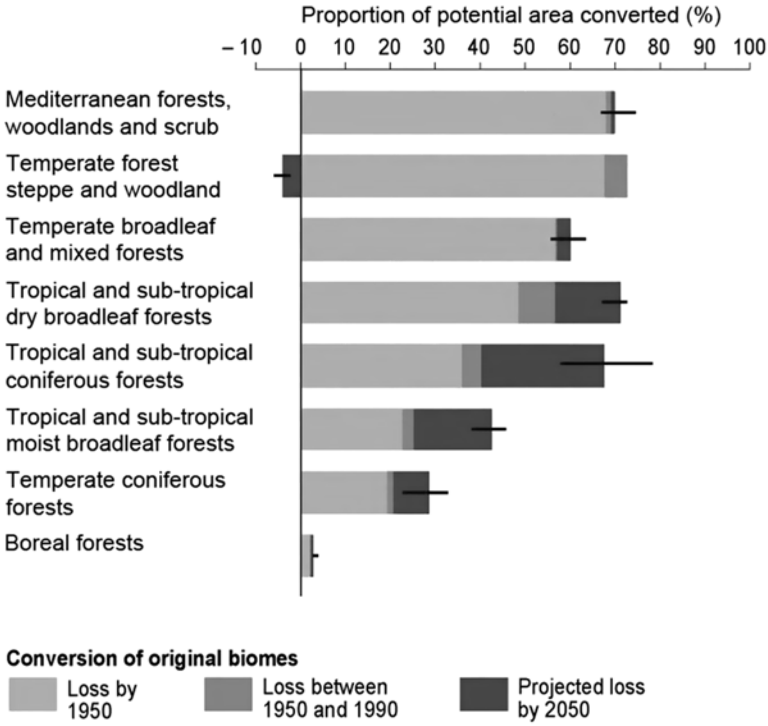

Figure 5.1 Past forest losses and projected future losses in the world's main forest systems. The proportion of the forest lost before 1950 was estimated based on the potential distribution of each forest system based on soil and climatic conditions. Projections of forest loss correspond to the average value of projections obtained for the four Millennium Ecosystem Assessment future scenarios; error bars indicate the range of values for the four future scenarios. Adapted from the Millennium Ecosystem Assessment (MEA 2005)

Globally, the overall extent of natural forests continues to decline, and the expansion of new forests does not compensate for the loss of natural ones (Butchart et al. 2010, FAO 2011). Moreover, the value lost with the degradation or deforestation of old-growth forests cannot be fully replaced by new forests because planted and regenerated forests differ from natural stands in many characteristics (Rey Benayas et al. 2009). First, restored forests do not support the same biotic communities as old-growth forests (Hobbs and Cramer 2007, Rey Benayas et al. 2009). Second, most new forests are located in temperate regions and cannot replace the biodiversity lost in highly diverse tropical regions; that is, the creation of forests in one region may not compensate for the destruction of forests in another region. Third, many planted forests are grown and managed for industrial purposes, so their contribution to biodiversity conservation and to the delivery of regulating and cultural services is modest or even negative (Kanowski 2003; Proença et al. 2010a, b). For instance, when continuous forest plantations replace traditional landscape mosaics, there is a loss of landscape heterogeneity and a decline, or even local

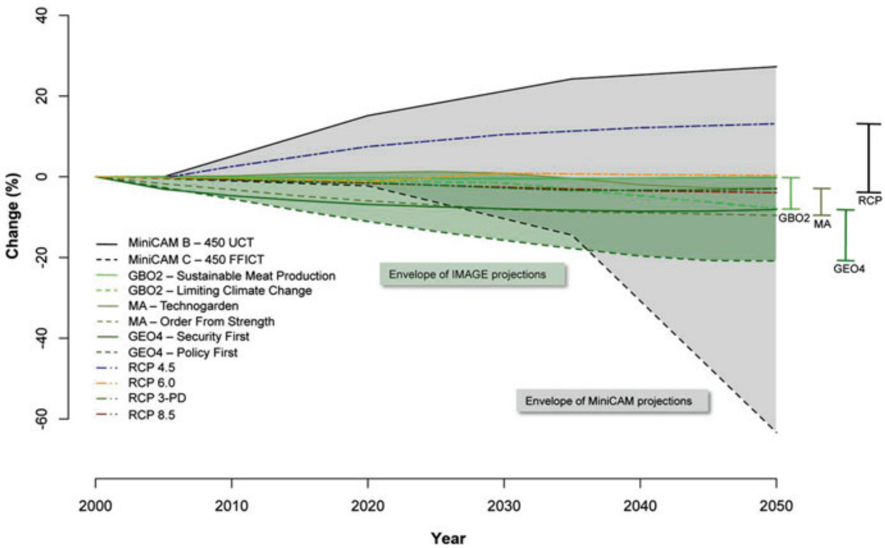

Figure 5.2 Projected changes in global forest cover until 2050 under various global scenarios: the Millennium Ecosystem Assessment (MEA) scenarios (Sala et al. 2005), the Global Biodiversity Outlook 2 scenarios (ten Brink et al. 2006), the Global Environmental Outlook 4 scenarios (UNEP 2007), the representative concentration pathway scenarios (Hurtt et al. 2009), and the MiniCAM scenarios (Wise et al. 2009). For each set of scenarios, we have only shown the two most contrasting results. The wider envelope for the MiniCAM projections, compared to the envelope of scenarios with the IMAGE model (IMAGE-team 2001), suggests that there are opportunities for action to reverse the global trend of forest decline, but also that wrong policy choices can exacerbate the loss of forest cover compared with the other scenario assessments. *Sources*: Leadley et al. (2010), Pereira et al. (2010)

extinction, of species associated with open habitats such as grasslands and meadows (Poyatos et al. 2003, Reino et al. 2009).

The global area of forest will probably continue to decline if countries fail to provide adequate incentives to halt deforestation. Global policy choices influence society's choices and may play a critical role in determining the selection of land-use options. A recent study by Wise et al. (2009) explored the effect of different carbon taxation policies on global land-use changes. The authors found that imposing a global carbon tax covering anthropogenic carbon emissions from all sectors, including emissions from land-use change, would promote the protection and expansion of forests, leading to an increase in forest cover (Fig. 5.2; the MiniCAM B scenario).

However, taxing only fossil fuel and industry emissions may prompt the expansion of biofuels and lead to a drastic loss of forest cover worldwide (Fig. 5.2; MiniCAM C scenario). Previous scenarios, which were also based on socioeconomic drivers, projected less drastic changes in forest cover (Fig. 5.2; MEA, GBO2, and GEO4 scenarios). The narrower range of variation in forest area projected by these scenarios is in part explained by compensatory mechanisms in the underlying

socioeconomic scenarios, which lead to a convergence of the trend lines for changes in forest cover area. For instance, the option for biofuels in the "environmentally friendly" scenarios implies the replacement of forests by biofuel crops (Leadley et al. 2010, Pereira et al. 2010).

Forest use and deforestation in the northern hemisphere are historically associated not only with socioeconomic development but also with ecosystem degradation, a shortage of forest products, and environmental disasters, such as flooding, which later motivated forest restoration and sustainable forest management in these regions (FAO 2012). Today, tropical forests are the ones most exposed to deforestation and forest degradation. The unsustainable use of forest resources jeopardizes socioeconomic development and human well-being in these regions (Rodrigues et al. 2009), and its negative impacts may also be felt at larger spatial and temporal scales (Leadley et al. 2010). Halting unsustainable use trends requires action from local to global levels and the adoption of socioeconomic development pathways that will ensure the sustainable use of forests, including the management of both tangible and non-tangible services, and sustainable support for human welfare. The management of forests and forest ecosystem services, and particularly non-provisioning services, should be based on polycentric and diverse governance systems that ensure an equitable representation of users and that promote knowledge sharing, collective decision-making, and enforcement of management decisions (Ostrom 2009).

5.3.2 Global trends in the use of forest ecosystem services

From a utilitarian perspective, natural forests have by default a multifunctional nature in the sense that they can simultaneously deliver several benefits to people, including forest goods (e.g., fuelwood, medicinal herbs, bush meat), regulating services (e.g., climate regulation, soil protection), and cultural benefits (e.g., esthetic pleasure, sacred groves). Still, some forests are designated and managed for a particular function, such as industrial plantations and forests in protected areas. Today, 30 % of the world's forests are primarily designated for productive functions (production of wood, fiber, biomass, and non-wood forest products), 24 % for multiple uses (forests managed to deliver a range of benefits without the dominance of a particular function), 11.5 % for the conservation of biodiversity, 8.2 % for protective functions (conservation of ecosystem functions and processes underlying the delivery of regulating services), and 3.7 % for social functions (recreation, education, and conservation of cultural heritage) (FAO 2011). The remaining 22.6 % are designated for other uses or remain unclassified.

The annual rate of growth between 2000 and 2010 was particularly high in regions with a low proportion of conservation forests compared with the global average (e.g., a 3.3 % increase in East Asia) but also in Europe (3.9 %, excluding the Russian Federation) and in South America (4.8 %). Most regions have already set aside 10 to 20 % of their forest for biodiversity conservation purposes (FAO 2011). The region with the highest proportion of conservation forest is Central America

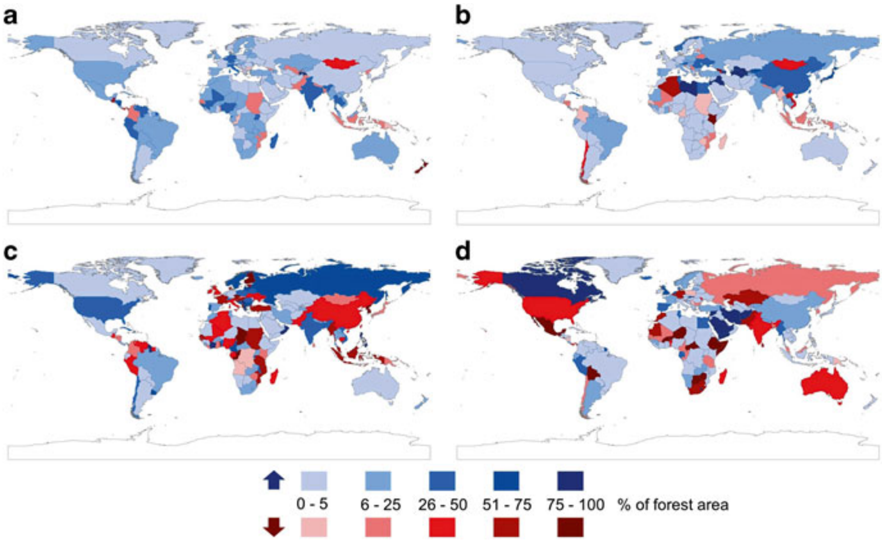

Figure 5.3 Proportions of the forest area designated for (**a**) biodiversity conservation, (**b**) water and soil protection, (**c**) production of forest products, and (**d**) multiple uses in 2010, and recent trends. Proportions are indicated by the color gradient, with darker tones indicating higher proportions. Trends are indicated by color, with *blue* indicating an increase in the designated area from 2000 to 2010 and *red* indicating a decrease. Note that the proportion does not indicate the extent (area) of forest in a country, and that the trends do not indicate the rate of change. *Source*: FAO (2011)

(47 %), whereas East Asia, West and Central Asia, and the Russian Federation have designated less than 10 % of their forest for conservation. Globally, 463×10^6 ha of forest have been primarily dedicated to biodiversity conservation (Fig. 5.3a).

Nevertheless, protective forests are now emerging as a tool to conserve ecosystem functioning and manage the delivery of regulating services. For instance, several Asian countries have reported a high proportion of protective forest (Fig. 5.3b). This is particularly the case for China, where a large area of forest has been planted with the main purpose of controlling desertification (Cao et al. 2011), and for several western Asian countries in arid zones, such as Turkmenistan and Uzbekistan, where water is a critical resource (FAO 2011). The same pattern is found for African countries in arid zones, such as Libya and Kenya.

Globally, the area of forest designated primarily for productive functions (Fig. 5.3c) has decreased at an annual rate of 0.2 % from 2000 to 2010, and currently covers 1200×10^6 ha (FAO 2011). This reduction in area is in part explained by the increase in the area of forest dedicated to intensive forestry but also because some areas previously classified as productive forests were reclassified as multiple-use forests (FAO 2011). Europe is the region that has reported the largest proportion of areas designated primarily for productive functions (52 % or 57 % excluding the Russian Federation), and North and South America have reported the lowest proportions

(14 %, but with heterogeneity within the region; for example, 1 % in Canada compared with 29 % in the United States) (FAO 2011). Despite the global decrease in the area designated for productive functions, the pattern is heterogeneous, showing a pattern of interspersed areas with increasing, decreasing, and stable trends.

In addition to areas specifically designated for productive purposes, many forest products are obtained illegally or informally from areas that are not classified as productive. This implies that the real area used for the extraction of forest products is much larger than the area that is formally designated as productive forest. Moreover, multiple-use forests (Fig. 5.3d) also encompass productive functions. Currently, the area designated for multiple uses totals 949×10^6 ha globally, and increased by 10×10^6 ha between 1990 and 2010 (FAO 2011). Global and regional trends are heterogeneous, reflecting different types of transitions, including shifts in the classification from productive to multiple use and vice versa but also shifts from undesignated to multiple use.

Social forests, which are primarily used for recreation, environmental education, and preservation of cultural heritage, are still infrequent, despite the widespread use of forests for outdoor activities and their cultural role as natural heritage sites. The social function of forests is usually associated with conservation forests or with multifunctional forests. Today, only 3.7 % of the world's forests are designated primarily for this purpose, but available data suggests that this proportion is increasing (FAO 2011). Brazil has the largest area of social forest, at 119×10^6 ha (i.e., more than 75 % of the global area of social forest), and this forest is designated for the protection of indigenous peoples and their culture.

The statistical data used in this section was reported by individual countries and gathered together for the Global Forest Resources Assessment 2010 (FAO 2011), which is the most comprehensive assessment to date. However, two main sources of uncertainty should be considered when comparing countries or regions. First, there are disparities among the reporting countries in terms of data availability, either due to real data gaps or due to differences in national forest inventory methodologies. Second, the criteria used to define forest categories and functions are subject to different interpretations by the reporting countries. Also note that the designation of a forest for a particular purpose does not imply the existence of sustainable management practices or even of a management plan for that forest.

5.3.3 *Drivers of change and impacts on forest ecosystem services*

The Millennium Ecosystem Assessment identified five main direct drivers of biodiversity and ecosystem change (MEA 2005): habitat change, climate change, invasive species, overexploitation, and pollution. The impacts of these drivers and their trends vary across the globe, and affect forest biomes differently. Direct drivers are often shaped by social demand for provisioning services, including both forest provisioning services and farmland services when agriculture replaces forest use.

For example, population growth can cause a higher demand for food and fiber and can therefore lead to production activities that cause deforestation. On the other hand, global policies for climate mitigation can encourage forest conservation, and technological advances can improve the efficiency of forestry and agricultural production, thereby lessening the pressure on natural forests.

Pollution became an important driver in the last century in many forms, and particularly in the forms of excessive nutrient loading in production systems and of industrial emissions (Shvidenko et al. 2005). Boreal forests have been particularly badly affected by air pollution from industrial sources during the last century, with reported events of significant tree damage and mortality (Shvidenko et al. 2005). When combined with climate change, pollution is expected to have a serious impact on the condition of these forests during the twenty-first century. Climate change not only will affect tree physiology and phenology but will also affect the fire regime by increasing the frequency and severity of wildfires as a consequence of drier and hotter summers (Soja et al. 2007, Stocks et al. 1998).

Habitat change and overexploitation were the main drivers of forest change in temperate regions during the last century. Today, the effect of these drivers is declining as new forests are planted and regenerate in abandoned fields and pastures. On the other hand, tropical forests have been particularly affected by land-use change during the last century, and the impact of this driver is expected to increase in the twenty-first century as forests are replaced by pasture and cropland (in part to respond to international demand for food) but also by infrastructure and urban areas. Overexploitation of forest goods is also expected to intensify due to population growth in these regions, as well as logging driven by international demand (Davidson et al. 2012, Lambin et al. 2003). Overall, the impacts of climate change will increase during the twenty-first century in all biomes (Leadley et al. 2010, MEA 2005). The impact of invasive species is also expected to increase due to global trade and travel, as well as the impact of pollution, in particular due to a significant intensification in the flow of reactive nitrogen into the environment (MEA 2005).

The effects of drivers are often synergistic. Changes caused by a driver or by a set of drivers may create the conditions for triggering, intensifying, or maintaining other drivers, rendering the control of their impacts difficult (Lambin et al. 2003, Leadley et al. 2010). In some situations, these interactions lead to regime shifts, with strong impacts on ecosystem structure and functioning. Although researchers can identify tipping-point changes and their potential risks, their dynamics are complex and difficult to predict (Leadley et al. 2010). Tipping points can be broadly defined as events that occur when an ecological threshold is passed, leading to shifts in ecosystem functioning that significantly affect biodiversity and ecosystem services. Tipping-point changes tend to be fast due to reinforcing feedbacks that amplify the effects of drivers or due to abrupt shifts when thresholds are crossed. They also tend to be difficult to reverse due to feedback loops that trap systems in undesirable stable states and long lag times between a driver's action and its impacts, which hamper policy decisions (Leadley et al. 2010, in press). The Amazonian, Mediterranean, and boreal forests present important examples of potential regime shifts in forest systems.

In the Amazon basin, forest conversion coupled with climatic changes may lead to a regime shift that will have impacts from local to global scales (Davidson et al. 2012, Nobre et al. 2010). Deforestation, logging, and forest fire are inducing regional climate changes, including less rainfall and increased frequency and severity of drought, which increase the susceptibility of forests to fire, thereby creating a feedback loop that sustains fire occurrence, promoting further forest damage and fragmentation. In addition, projections from climate models indicate that long-term global climate change will amplify drought in the Amazon region due to a combination of climate warming and less precipitation. At moderate to high rates of deforestation, the interaction between land-use change, fire, and climate change may lead to a feedback loop that will be difficult to control and that may cause extensive forest loss (Davidson et al. 2012, Vergara and Scholz 2011). Carbon release due to this deforestation will also contribute to global climate change and will aggravate climate-change impacts at the regional scale.

Consequences for ecosystem services and biodiversity will be severe. The Amazon is one of the world's largest carbon pools and carbon sinks, with the exception of dry years, when forests becomes a carbon source (Davidson et al. 2012, Phillips et al. 2009). The shift from a carbon sink to a carbon source will contribute to global warming and cause negative impacts at a global level. The Amazon is also a biodiversity center (Pereira et al. 2012), and loss of Amazonian forest will result in a severe loss of biodiversity at a global scale. In addition, there is the risk of losing species, many still unknown, that have medical and pharmacological value, and consequently a risk of losing the opportunity to find and develop new medications and vaccines. At local and regional scales, local communities will be affected by the loss of forest goods, including food, fiber, fuel, and medicinal plants; by the loss of regulating services, such as climate regulation, fire regulation, and flood regulation; and by the loss of cultural services, since the forest environment is a major component of the cultural heritage and way of life of local peoples.

In the Mediterranean region of southern Europe, land-use change, fire disturbance, and climate change are interacting to create conditions suitable for a shift in ecosystem composition (Proença and Pereira 2010). Rural abandonment is driving land-use change in marginal areas of farmland, through the regeneration and encroachment of natural vegetation and the expansion of fire-prone forest plantations, thus promoting fuel continuity in the landscape. The accumulation of biomass coupled with frequent (anthropogenic) fire ignition is causing a change in the fire regime, with more frequent and severe fires. This situation is further aggravated by climate change, in particular by hotter and drier summers. All these factors cause an increase in fire risk and promote the expansion of fire-prone communities, such as shrublands, which then create the conditions for the establishment of a feedback loop that inhibits the progression of natural succession towards regeneration of natural forests. Under some circumstances, alien invasive species gain competitive advantages in the burned areas, letting them replace native species and impoverishing natural communities (Keeley et al. 2003, 2005). This may eventually lead to a compositional shift that will be hard to reverse.

Boreal forests provide a third example of regime shifts. In this case, climate change is leading to a warming trend that is moving northward, creating an environment unsuitable for boreal species. On the one hand, these species may not be able to respond to this change because their natural rate of propagation is too slow for them to migrate north and also because tundra sites into which the boreal species will be forced to migrate might be unsuitable for their establishment and growth (Lloyd et al. 2011, Soja 2007). Chapter 2 of this book discusses these issues in more detail. But on the other hand, where tundra sites are suitable for boreal species, changes in soil albedo, the melting of snow cover, and retreating permafrost will allow tree establishment and forest invasion into the tundra (Soja 2007). The main mechanism underlying this change is an amplifier feedback loop driven by increasing summer temperatures (Fernandez-Manjarrés and Leadley 2010). Warmer temperatures lead to earlier melting of the snow cover, exposing soil with a lower albedo that traps more solar radiation over a larger period. Snow cover creates an insulating effect that is critical for the maintenance of permafrost; the loss of snow cover causes permafrost degradation and increases the warming effect, thereby promoting further snow melting. These changes will have impacts on the lives of local people, who will have to adapt to a changing landscape (e.g., travel routes may become unsafe due to decreasing ice stability), but will also have impacts at a global level due to the release of large quantities of carbon and methane stored in the permafrost, with consequences for climate change (Fernandez-Manjarrés and Leadley 2010, Schaefer et al. 2012). Moreover, some parts of the boreal forest are being increasingly affected by fires driven by climate change, which also causes the release of carbon stored in the trees and soil in addition to the loss of old-growth forest and other social and ecological losses (Soja et al. 2007).

As the demand for non-provisioning services (e.g., soil protection from erosion, water purification, recreation) increases, it counteracts the economic bias towards provisioning services. Direct drivers will be gradually affected by indirect drivers in response to this demand. This may include national to global policies but will also include changes in markets. In the past, and to a certain extent, still today, economic choices tended to disregard non-market services and promote the expansion and overexploitation of productive forests or the replacement of forest by more profitable land uses. The incorporation of the benefits delivered by non-provisioning services in economic choices is likely to reshape market demand and its effect on the direct drivers of change, thereby promoting forest conservation and restoration to preserve forest's regulating services and cultural value.

5.4 Economic valuation of forest ecosystem services

Forests can provide multiple benefits to society other than wood, with the whole array of benefits depending on the characteristics of the forest and the prevailing management strategies (Duncker et al. 2012). This understanding is a prominent feature of the current literature and is usually associated with the concept of

multifunctional forests (e.g., Carvalho-Ribeiro et al. 2010, Gustafsson et al. 2012). One possible approach to capture the contribution of forest ecosystems to humans is through an improved understanding of the linkage between the functioning of the ecological system, which is perceived as a composite of processes and structures, and the functioning of the socioeconomic system. The crucial role that natural systems play in underpinning economic activity and human well-being is of growing concern (Bateman et al. 2010). Thus, economic valuation of ecosystems and their services has been receiving increasing attention in the literature.

Economic valuation is not the only approach to assigning a value to nature, nor is it necessarily the best approach; for examples of other forms of valuation, see Oksanen (1997), Martín López et al. (2012), and TEEB (2010). As Kareiva et al. (2011) pointed out, it is important to emphasize that an economic valuation does not replace or ignore the intrinsic value of nature, nor does it reduce the moral imperative to conserve nature. Following the logic of Martín-López et al. (2009) and Mace and Bateman (2011), we note the importance of combining economic and other valuation approaches to provide a more holistic picture of the value of forests. Nonetheless, in this chapter we will focus on the economic valuation approach. The primary role of economic analysis is to assist decision-making (Daily et al. 2000, Pearce et al. 1989, Tietenberg 1996). In the context of forest management, the high rate of deforestation we are facing globally—13×10^6 ha per year (FAO 2007)—and the rise of international concern about the consequences of deforestation together mean that economic valuation of forest ecosystem services has an important role to play.

Before jumping into the principles and methodological details of economic valuation, we will briefly illustrate how economic valuation of a forest ecosystem can restrain deforestation. As we noted earlier, forests provide many non-market goods, such as watershed protection. Landowners seek profit maximization, and in the absence of other mechanisms, they rely on existing markets to pursue this goal. Existing markets define their costs and revenues. Hence, even though we know that clearing the forest would increase problems such as downstream flooding and sedimentation, these costs do not accrue to the landowner who will decide whether to harvest the forest; thus, these costs are not factored into the landowner's decision. This is clearly a market failure from a larger perspective. Economic valuation can mitigate this problem if the analysis allows for an extended accounting of benefits and costs and, based on this more complete picture, fosters mechanisms such as subsidies, taxes, direct payments, and payments for ecosystem services that can prevent the market failure and reduce the likelihood of deforestation. For a concise review of market-based mechanisms, see Pagiola et al. (2002). These mechanisms aim to fully internalize the benefits and costs that do not accrue directly to landowners but rather that affect other groups in society. In Sect. 4.1, we further explain the occurrence of externalities and market failure from a conceptual point of view.

At this point, and before we begin discussing the principles of economic valuation, we want to emphasize that the value of forest ecosystem services reflects the different ways in which they satisfy human needs. This can be considered from the perspective of the total economic value (TEV) taxonomy (Pearce 1993).

This taxonomy defines the different sources of values that people may attach to the different services provided by a given ecosystem. Note that this taxonomy relies on whether ecosystem services satisfy human needs directly or indirectly. Economic value, then, is a measure of the degree of satisfaction provided by these services. The TEV approach and terminology are not uniform across the literature, but TEV generally includes the following value components: direct use, indirect use, option, and non-use. The first three categories are generally referred to together as use values, and the non-use values often aggregate values such as bequest and existence values.

Among the use values, direct use values include services that are used directly, and include provisioning services (e.g., forest goods) and cultural services (e.g., recreation opportunities). Indirect use values include services that are indirectly used, such as the benefits derived from regulating services (e.g., climate regulation). Non-use values are divided into bequest and existence values, and are almost entirely associated with cultural services. Bequest values represent the value that an individual assigns to an ecosystem or species due to its relevance to the well-being of future generations. Existence value, on the other hand, represents the value that an individual assigns to an ecosystem or species due to its personal relevance at the present time. In other words, it is the satisfaction this individual derives from knowing that a certain species or ecosystem exists.

Option values include all values (both use and non-use) that are expected to be enjoyed in the future (e.g., provision of genetic resources, maintenance of a gene pool for bioprospecting, cultural heritage). Note that the option and bequest values both reflect the importance that people give to maintaining or restoring ecosystems in order to ensure the delivery of ecosystem services in the future.

5.4.1 Principles of economic value estimation

The economic value of an ecosystem service refers to the contribution of a certain ecosystem functional dynamic to human well-being. Many ecosystem services are only obtained because of other capital inputs; for instance, agricultural production of food implies the use of machinery and labor together with the use of natural resources and ecosystem processes. Hence, as pointed out by Bateman et al. (2010), estimating the economic value of ecosystem services requires isolation of the ecosystem function's contribution before the value can be converted into a monetary metric. This suggests that it is also necessary to clarify how economic analysis differs from financial analysis: the former examines society as a whole, whereas the latter focuses on particular groups within society. Hence, when estimating the economic value of an ecosystem service, we must account for the costs (private and external) of producing the service and for the benefits (private and external) generated by it. Here, "external" refers to externalities, whether benefits or costs, that are generated as unintended by-products of an economic activity that do not accrue to the parties involved in the activity, and for which no compensation is provided.

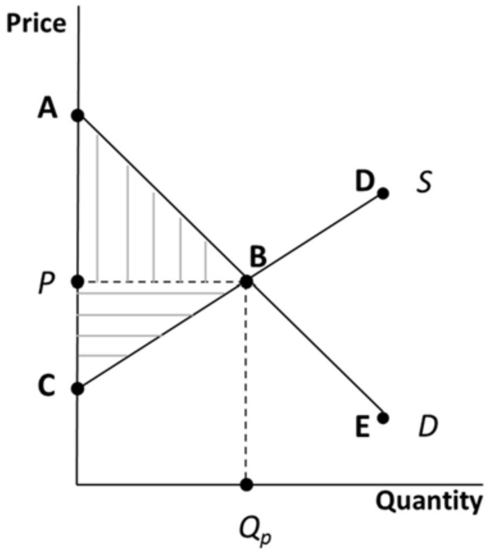

Figure 5.4 Social surplus [*ABC*] for a forest good such as timber under perfect market conditions. *D* demand curve, *S* supply curve, *P* price where *D*=*S* at point *B*, Q_p the quantity at price *P*

Depending on its impact on a third party, an externality may be positive (e.g., the creation of a forest landscape) or negative (e.g., the creation of fragmentation).

We first approach the economic foundation of ecosystem service valuation by considering a well-defined market in which ecosystem services can be traded and in which there are no external costs or benefits. There are two building blocks in the process of estimating economic value: consumer and producer surpluses, with the social surplus equaling the sum of these two surpluses. See Mankiw (2008) for a discussion of this topic. These measures are illustrated in Figure 5.1 for the case of an ecosystem service for which there is a market, such as timber (a forest provisioning service), based on the assumption of a perfectly competitive market. The timber market is in equilibrium when demand (*D*) equals supply (*S*) at price *P*. The demand curve shows consumer marginal willingness to pay (WTP), which represents the consumer's WTP for each additional unit of a product. The supply curve shows the marginal costs of harvesting timber, which represents the producer's marginal willingness to accept (WTA) a given price for their product.

Figure 5.4 tells us that buyers who value the good more than the price (represented by the line segment *AB*) choose to buy the good and receive a surplus of benefit: the area of the triangle *ABP* defines the magnitude of the consumer surplus. This represents the amount a buyer is willing to pay for a good, minus the amount the buyer actually pays for it, or, in different words, the benefit that buyers receive from participating in the market. Buyers who value the good less than the price (represented by the line segment *BE*) choose not to buy the good or receive its benefits. Symmetrically, on the production side, those sellers whose costs are less than the price (represented by the line segment *CB*) choose to produce (in this case, to harvest) and then sell the good (wood). Sellers receive a surplus given by the area of

the triangle *PCB*; this represents the amount a seller is paid, minus the cost of production. The producer's surplus measures the benefit sellers receive from participating in the market. Sellers whose costs are greater than the price (represented by the line segment *BD*) do not sell the good or receive benefits from the sale. The social benefit (i.e., the overall surplus) in this case equals the private benefit, which equals the sum of the consumer and producer surpluses (i.e., the area of triangle *ABC*). The social surplus is of interest in economic analysis because it concerns the net benefits that society as a whole derives from the good. Mathematically, the total or social surplus can be expressed as follows:

$$\text{Social surplus} = \begin{bmatrix} (\text{value to buyers} - \text{amount paid by buyers}) \\ +(\text{amount received by sellers} - \text{costs beared by sellers}) \end{bmatrix}$$

Thus far, we have analyzed a situation in which the social benefit equals the private benefits. However, when there are external costs and benefits, the private surpluses do not equal the social surpluses. Let's again consider a timber market, but in the presence of external benefits, based on the example provided by Hanley and Barbier (2009). Consider a sustainable timber harvester, with sustainability here defined as a state of non-declining well-being, as defined by Tietenberg (1996; pp. 33–34). This harvester manages their land in a wildlife-friendly manner, thereby improving the ecological quality of their woods and overall forest health by (among other things) creating many habitats for birds and butterflies. They also harvest timber for sale. The market rewards them for their timber production, since they can sell the timber to interested buyers. But the market is unlikely to reward them for their "production" of wildlife habitats, even though these habitats might be valued by society. Although this is not the forum for further discussion of this topic, these types of services fall into the category of public goods (non-excludable and non-rival in consumption). See Boardman et al. (2001) for further explanation. In Figure 5.5, D_m is the market demand curve for timber, *S* is the supply curve for

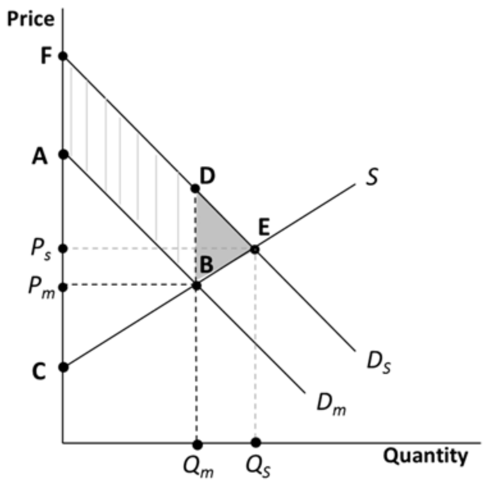

Figure 5.5 Social surplus (*ABDF*) for a good (e.g., timber) in the presence of a positive external effect (e.g., provision of wildlife habitat). *S* supply curve, D_m market demand for timber, D_s society's demand for timber plus its external benefits, Q_m the quantity at price P_m, Q_s the quantity at price P_s. *Grey area* (*BDE*) represents welfare lost under perfect market conditions (e.g., without government intervention)

timber, and D_s is society's demand for timber plus its external benefits (in this example, "production" of wildlife habitats). The market reaches equilibrium at point *B*. At this point, both timber consumers and the rest of society receive a benefit (an external benefit whose magnitude equals the vertical distance between D_m and D_s) which is represented by the line segment *BD*. The social surplus obtained when quantity Q_m is produced therefore equals the area of the polygon *ABDF*. Notwithstanding, the social optimum would be reached at point *E*, where the marginal social cost (which, as no negative externality is being considered, is the same as the marginal private cost) is equal to the marginal social benefit. Under perfect market conditions (e.g., without government intervention), quantity Q_s will not be supplied because harvesters are not rewarded for producing such quantity. The area of triangle *BDE* (grey shade) represents the welfare loss under these market conditions. Many ecosystem services are externalities, in the sense that the benefit or cost they represent to society is generated as a consequence of standard ecosystem management but it is not intentionally produced, and it does not accrue benefits or costs to the producer. This means that, for instance, the value of timber does not reflect the array of benefits that may be jointly provided by forests to society as a whole. Often, in the literature, these externalities are referred to as non-market ecosystem services. As we discuss in the next section, several methods have been developed to estimate the value of such ecosystem services.

5.4.2 Economic valuation methods

Our purpose is not to fully review all the available valuation methods, but rather to provide a concise overview of such methods while illustrating the objective and context of their application. In the previous section, we focused on consumer and producer surpluses as the measures of interest and explained how these measures relate to WTP and WTA. Bearing in mind that these are the measures of interest, we should also note that the focus of economic valuation is to estimate such measures for a well-defined change. This implies estimating the changes in the consumer and producer surpluses and the change in their sum (Freeman 2003) by considering changes in the welfare of both consumers and producers.

The methods used to value ecosystem services can be grouped into three main categories: direct market valuation approaches, revealed preferences approaches, and stated preferences approaches. Direct market valuation approaches rely on the use of data that can be readily obtained from existing markets (such as prices, demanded quantities, and production costs), and include three main approaches: approaches based on market prices, costs, and production functions. Market price approaches rely on the use of market prices as a proxy for value. Although this appears to be the most straightforward approach, there are several aspects that should be emphasized about its application. Under the general case of perfect competitive markets, prices are defined by the interaction of supply and demand; as a result, prices are acceptable or starting point approximations of the marginal value.

If this holds true, and the change being analyzed is sufficiently small that prices remain constant, application of the method is straightforward: we just multiply the change in the number of units (for instance, the increase or decrease of the available m^3 of water) by the associated marginal price. When the changes are large enough to change prices, then the changes in consumer and producer surplus must be estimated. Even if prices can be taken as a proxy for the marginal value, price distortions created by subsidies and taxes should be taken into account; the cost of making the good available should be subtracted from the price in some cases, since labor and transportation costs involved in making the benefit available represent opportunity costs that could be transferred to generate alternative goods and values; in addition, prices generated by supply and demand reflect scarcity, not value, as is often illustrated using the relative prices of water and diamonds (a paradox originally posed by Adam Smith), since water is vital to support life (unlike diamonds) but because it is generally abundant, it is cheaper than diamonds.

Cost-based approaches include the avoided cost, replacement cost, and mitigation or restoration cost methods, and are used to estimate the costs that would be incurred to artificially provide the benefit instead of using ecosystem services. In the context of forest ecosystem services valuation, the avoided cost method could be used (for example) to estimate the value of flooding protection provided by a forest based on the costs of building protection infrastructures to generate the same benefit; for other applications of the avoided cost method, see Nowak et al. (2006) and van Kooten (2007). The replacement cost method could be applied (for example) to estimate the value of soil protection based on the costs to restore the storage capacity of downstream dams after siltation of the reservoir. For other applications of this method, see Chopra and Kumar (2004) and Rodríguez et al. (2007). The restoration costs may, for instance, be useful in determining the value of water purification or infiltration based on the investments made to reverse degradation of the service. For other examples of the restoration cost approach, see Birch et al. (2010).

The last of the approaches based on market valuation is the production function method, which Barbier (2007) referred to as "valuing the environment as input". Behind the method's application is the idea that several ecosystem services (e.g., regulation services, biodiversity) enhance the production of market goods. Hence, if changes in these services affect the marketed product, then the effects of these changes will be visible through the price system. For instance, if the purification capacity for water decreases and this generates additional costs for the producers of bottled water, then the price of the water would increase. An example of this method in the context of forest ecosystem service valuation is provided by Nahuelhual et al. (2006).

In the revealed preferences approach, the main methods are the travel cost and hedonic pricing methods. These methods use consumption behavior in markets that are related to the non-market goods and that therefore serve as proxies for those goods. The travel cost method is the most commonly used method, and has been widely applied to infer the value of forests for recreation (e.g., Badola et al. 2010, Bowker et al. 2007). This method uses visitation rates and the distance traveled to infer the demand for such a benefit. The observed variation in visitation rates and

travel costs (used as a proxy of price) describes the changes in demand for the site, and the demand function allows researchers to determine the consumer surplus. The hedonic pricing method uses the differences in the price of a benefit that reflect its inherent properties to infer the value of non-market attributes of ecosystems. A recent application of the method was provided by Sander and Haight (2012).

In the stated preferences approach, contingent valuation is the most well-known method. This method involves directly asking a representative sample of a population to define their WTP and their WTA for a well-defined change in the provision of a certain ecosystem service (for instance, a change in water quality). Researchers can then use compensating variation or equivalent variation to estimate the economic value. Both are exact welfare measures, and may not be identical to the consumer surplus for market goods. Instead, these measures estimate the change in income that is needed to maintain a certain level of utility (welfare, satisfaction). Note that along an ordinary demand curve, utility is not constant if income is kept constant. For further explanation of these measures, see Freeman (2003) and Zerbe and Bellas (2006). Choice modeling is a questionnaire-based method that gained relevance with practitioners of economic valuation of ecosystem services. The method consists of presenting individuals with two or more alternatives defined by a set of attributes regarding the ecosystem services under valuation, and it is designed to elicit the WTP for having that alternative. The levels of the set of attributes vary among the alternative sets that individuals must choose among or rank. Both methods have been applied to estimate the value of several forest ecosystem services. For examples of contingent valuation applications, see Sattout et al. (2007) and Barrio and Loureiro (2010); for examples of choice modeling, see Rolfe et al. (2000) and Brey et al. (2007). Individual-based questionnaires aggregated to represent a socially relevant unit (e.g., a community) might be appropriate when the services being valued are purely enjoyed on an individual level (e.g., valuing forests for timber), but have limited applicability in the cases of more communal services. For example, the value of forests to a community whose social system is intimately dependent on them is more than the sum of the independent personal values (Farber et al. 2002). Hence, another stated preferences method, group valuation, is gaining relevance. Although a stated preferences method, its focus is not on valuing individual preferences but rather on collecting social preferences. Wilson and Howarth (2002) and Chan et al. (2012) provide a detailed discussion of this method.

5.4.3 Challenges in estimating economic value

Estimating the economic value of ecosystem services faces several challenges, and regardless of the objective of the economic valuation, whether to inform macroeconomic policies or to evaluate programs (Bateman et al. 2010), estimation of flows of ecosystem services is often necessary. A flow estimation is usually an estimate of money per unit area obtained for a certain period, usually on an annual basis. Although flow estimations provide valuable information, they are not, per se,

relevant to inform land-use decisions because few interventions would result in an entire loss of the flows of ecosystem services. Instead, management often results in incremental small changes. What is needed is an understanding of how land-use changes would affect societal well-being, so the focus of the economic valuation is on valuing the incremental or marginal changes in the flows of services. This is often done by means of scenario analysis, in which researchers compare the consequences of two or more scenarios. Valuing such changes implies a deep understanding of the ecological dynamics of the system and how the system responds to perturbation.

Though the economic valuation approach is remarkably valuable because of its ability to provide more objective comparisons of alternatives, it has not yet overcome significant challenges to tackling such complexity (Robertson 2011). This problem has been pointed out by several authors under the headings of uncertainty, ecological thresholds, and irreversibility (Morse-Jones et al. 2011), and in the contexts of weak or strong sustainability (Olschewski and Klein, 2011). Because the valuation focuses on estimates of marginal changes, caution is needed with the valuation itself because the marginal value may not be constant. This is clearly illustrated by the example provided by Bateman et al. (2010), who examined the recreational value of an urban green space (a park). They found that increasing the area of this space altered the recreational marginal value, with the first increases in area being highly valued, but subsequent increases becoming less valued.

There are other problems related to the assumption of a constant marginal value that suggest a need for caution. Ecosystems and their services are not spatially homogeneous and thus may not provide the same flow throughout the system's spatial extent (Fisher et al. 2009). Moreover, even when ecosystems provide the same flow of services from different areas, the marginal values of these flows may not be the same. We can illustrate this again using the value of a green space for recreation. An urban forest area of a given size may have a higher recreation value when it is near an urban area than when it lies in a region that is not accessible to urban residents. The issue of spatial variability of ecosystems and ecosystem services suggests the need to perform economic valuation on a spatially explicit basis. In addition, the effect of scale is a challenging topic that has not been fully tackled. This affects the discussion of ecosystem services valuation because the scale at which benefits might be provided ranges from local to global. For example, a forest might provide recreational opportunities (local), downstream flood prevention (regional), and climate regulation (global) when considered from the supply side. This variability also holds for the demand side. For instance, endemic species may have beneficiaries very far from the location of their occurrence (as in the case of residents of developed nations placing value on endangered species in developing nations). The issue of scale has been extensively debated (EEA 2010, Hein et al. 2006), and the advantages of spatial analysis in tackling the issue are making scale-explicit analyses increasingly relevant. Failing to properly address the issue of scale may complicate or bias the design of ecosystem services payments, which is an emerging mechanism to ensure the provision of non-market ecosystem services.

5.4.4 *Economic estimates of forest ecosystem services around the world*

As we have stressed in the previous sections of this chapter, awareness of the importance of forest ecosystems and the vulnerability of their valuable services is increasing around the world. This awareness is ultimately at the base of recent economic valuation efforts that targeted forest ecosystem services. In this section, we provide an overview of recent studies in which forest ecosystems from around the world were monetarily valued.

Previously, we briefly introduced the concept of TEV as a commonly used taxonomy to determine the aggregate economic value of all the benefits people obtain from ecosystems. Several researchers have attempted to obtain the TEV of forests (e.g., Adger et al. 1994, Merlo and Croitoru 2005, Pearce 2001, Thompson et al. 2011) and Ferraro et al. (2012) provide a review of this topic.

Targeting specific areas for the estimation of TEV is a common practice. One recent attempt to assess TEV focused on the Hoge Veluwe Park (Hein 2011), a protected area in the Netherlands. Hein estimated the economic value of ecosystem services provided by the park's more than 5000 ha of pine and deciduous forests to be around 10.7 million € per year, of which 2.1 million € per year were due to air pollution removal and 1.9 million € per year were due to groundwater infiltration, for example. By combining land-cover mapping with benefit-transfer calculations, Vorra and Barg (2008) estimated the aggregate economic value of ecosystem services provided by the Canadian UNESCO World Heritage Site of Pimachiowin Aki to be approximately C$130 million per year, mostly due to its provision of pure water and fish. Ingraham and Foster (2008) used a similar approach, but their goal was even more ambitious: to determine the aggregate economic value of the ecosystem services provided by a large network of protected forests (the U.S. National Wildlife Refuge System) in the 48 contiguous states, which turned out to be around $26.9 billion per year. Although this was a first approximation, their research highlighted the need for further and more rigorous examination of the value of ecosystem services to assist management and policy decision-making, and their results emphasized that the TEV of protected areas exceeds that of its pure recreational value.

Bolder initiatives to assess the aggregate monetary value of global forest ecosystem services have also taken place in recent years. By using ad hoc value-transfer protocols, Chiabai et al. (2009) estimated the economic value of a comprehensive set of ecosystem services (timber and non-timber forest products, carbon storage, and recreation and tourism) for all forest biomes around the world. Benefit (or value)-transfer uses economic information captured at one place and time to make inferences about the economic value of environmental goods and services at another place and time, as discussed by Wilson and Hoehn (2006). An interesting aspect of this research is that it also presents potential estimates of total economic losses by the year 2050 due to policy inaction. They identified major economic losses of 78 billion €, mostly due to the loss of the forests' provisioning and carbon sequestration services. Their results underlined current production scenarios; for example, they

estimated the marginal value of provisioning services from tropical forests in Africa at around US$1800 per ha per year as a result of the high production value of fuelwood in Africa.

Even though valuing nature as a whole through the TEV taxonomy might be pertinent for current conservation agendas, valuation studies are most commonly performed on a case study basis, with a particular ecosystem service or a particular subset of services being targeted. Focusing on a particular ecosystem service might help to solve specific challenges and support policy development at a relatively small scale, as it is a far simpler task than estimating TEV and may provide more practical outputs. For example, to provide insights into the protection of woodlands as a climate-change mitigation measure, Brainard et al. (2009) performed a cost–benefit analysis to assess the value of the carbon sequestration services provided by the woodlands of Great Britain. Their results, which depended strongly on the discount rate and the social value of sequestered carbon being considered, ranged from US$82 to US$853 million annually. The variation of estimations due to the application of different discount rates is not uncommon, and imposes an additional challenge for the valuation of nature (Freeman and Groom 2013). In fact, the choice of the discount rate to be applied in environmental appraisal is not exempt from problems and criticisms, and there is no single discount rate to be applied, so sensitivity analysis may be generally performed for a range of discount rates (Boardman et al. 2001).

Appraising the value of the carbon sequestration services of forests as part of the tactics for mitigating human-origin CO_2 emissions has been receiving increasing attention in response to growing awareness of climate change. In Brazil, Guitart and Rodriguez (2010) have assessed the value of potential carbon sequestration services provided by two commercial eucalyptus plantations. Based on their results, they suggested a minimum annuity of US$18.8 per ton of stored carbon to be paid to the owner of the forests in order to stimulate and justify the adoption of silvicultural regimes that would increase carbon sequestration.

Though globally relevant nowadays, the ability to sequester carbon is not the only valuable service that forests can provide. Olschewski et al. (2012) determined the WTP for the avalanche protection services that the forests of the Swiss Alps provide to the population of the Swiss municipality of Andermatt, in Canton Uri. Their results ranged from US$20 per household (a one-time payment due to avoidance costs) to more than US$300 per household (a one-time payment due to risk reduction). In Africa, the risk of snow avalanches is obviously not a primary concern. Instead, Schaafsma et al. (2012) estimated the total flow of benefits from charcoal production in the tropical forests of southern Kenya to be worth around US$14 million per year, as charcoal provides an important source of income to local households and supplies around 11 % of the charcoal used in the major cities of Kenya and Tanzania. For timber forest products, Ojea et al. (2012) estimated the potential value of wood provision in sustainably harvested Mediterranean forests in Spain at around 500 € per ha per year based on the sustainable provision of timber over 30 years at a discount rate of 2 %. Their results indicated that, in some cases, non-sustainable forests provided higher returns over shorter time spans, but that sustainable

harvesting provided the highest overall returns over long terms. Regardless of the ecosystem service targeted, studies at a regional level bring a very policy-oriented perspective into the valuation exercise.

5.5 Initiatives and policy responses

Today, most large-scale initiatives and agreements regarding forest management revolve around the concept of sustainable forest management to prevent forest degradation and to promote the development of multifunctional forest systems.

The REDD+ mechanism (*Reducing emissions from deforestation and forest degradation in developing countries and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries*) is currently the most promising tool designed to support the conservation of forests, with a particular emphasis on carbon-regulating services (FAO 2012, http://www.un-redd.org/). The mechanism was established under the United Nations Framework Convention on Climate Change (http://unfccc.int/2860.php) and included in the global climate-change agenda in 2007 that was defined at the climate summit in Bali (Angelsen and Rudel 2013). The mechanism has been implemented through several initiatives, such as the UN-REDD program (http://www.un-redd.org/) or the Forest Carbon Partnership Facility hosted by the World Bank (http://www.forestcarbonpartnership.org/). REDD+ is a financial mechanism designed to reduce carbon emissions caused by forest losses and degradation in developing countries while at the same time creating conditions for sustainable forest management and promoting sustainable development programs in the participating countries (Angelsen et al. 2012, IUCN 2009). In brief, the underpinning idea is to implement payment for ecosystem services (PES) schemes in which the international community pays forest users in developing countries to adopt policies and programs aimed at conserving forests, improving forest stocks, and reducing forest degradation (Angelsen et al. 2012).

Despite the overall support for this mechanism, and its ongoing application in several countries, it has also been criticized based on several important issues. These include the need to guarantee sustainable sources of funding, the difficulty in monitoring the outcomes of implemented projects, and more importantly, the lack of a clear understanding and a legal framework for land tenure and carbon rights in many countries, which can be a barrier to the implementation of PES schemes, particularly if this promotes inequity and disregard for the rights of forest communities and indigenous peoples (Angelsen et al. 2012, FAO 2012). In addition, there is some apprehension concerning the subordinate role of biodiversity in relation to carbon storage and sequestration, which constitute the main focus of REDD+. Unclear targets for biodiversity and other ecosystem services may allow the occurrence of trade-offs instead of achieving the envisioned synergies, which are expected to arise as positive externalities from activities directed towards carbon storage and sequestration (Visseren-Hamakers et al. 2012). For instance, there is a risk of leakage (i.e., intensification of activities in areas not covered by REDD+ projects) and of inadequate

implementation of REDD+ activities, such as the establishment of plantations of exotic species (Visseren-Hamakers et al. 2012). Also, because there is some disconnection between the global distribution of carbon stocks and the associated biodiversity, the outcomes of REDD+ projects may be less effective for protecting biodiversity and other ecosystem services (Visseren-Hamakers et al. 2012).

Other PES mechanisms are emerging at a regional scale and are aimed at conserving forest ecosystem services. For instance, in the European Union, under the current European Agricultural Fund for Rural Development (http://ec.europa.eu/agriculture/cap-funding/budget/index_en.htm), a payment scheme has been implemented to support the development of multifunctional forests and the adoption of good management practices.

The rationale behind PES mechanisms lies in their attempt to internalize market externalities. As described above, many ecosystem services are not tradable in a market; thus, producers are unable to introduce their value into the price of the products they supply. Due to this market externality, producers will always be underrewarded for the services they provide in the absence of production incentives or subsidies. Through PES, central governments or private users and consumers make payments for the ecosystem services provided by landowners, producers, and other entities such as environmental agencies or nongovernmental conservation organizations. Examples involving the industry sector reflect how industries can often play a leading role as beneficiaries or buyers of ecosystem services, as in the case of the Nestlé Waters Programme (www.nestle-waters.com/).

Although the EU Forest Strategy is implemented at a regional scale, it nonetheless provides a good example of intergovernmental action to promote and support sustainable forest management through the coordination of forest policies by the member states and through community policies (CEC 2005). The strategy also acknowledges the multifunctional role of forests and their multiple services, and their relevance for the well-being of society. The Ministerial Conference on the Protection of Forests in Europe (http://www.foresteurope.org/) constitutes the political process at a pan-European level for the establishment of sustainable forest management. The Forest Europe strategy for 2020 developed by Ministerial Conference on the Protection of Forests in Europe has been signed by 46 countries, and lists among its main targets the valuation of multiple forest services and raising society's awareness of the importance of forests to human well-being. This strategy will foster cooperation among countries to develop and update their forest policies so as to secure and promote sustainable forest management.

Targeting different scales, the certification or information labeling of agroforestry products might also be a way to communicate environmental and other attributes that are not directly visible in the products, with the goal of promoting sustainable management of forestry resources. The rationale behind certification mechanisms is simple: if there is a market demand for differentiated agroforestry products, meaning that consumers are willing to pay for the price difference listed by producers due to their compliance with environmental standards and the consequent delivery of external benefits, then a market-based solution for sustainable rural development becomes possible. Countries like the Netherlands, Germany, and

the UK are clear examples of major markets for certified forestry products, and specifically for timber. The most widely acknowledged example of certification mechanisms are the regional and national standards developed by the Forest Stewardship Council (https://ic.fsc.org/). Other less widely known initiatives include the Programme for the Endorsement of Forest Certification Schemes (http://www.pefc.org/), which is now a recognized label in Europe and in a few other countries such as Brazil and the United States. FAO (2007) estimated that around 270×10^6 ha of forests around the world, amounting to roughly 7 % of the world's forests, are certified for sustainability through an independent labeling organization. However, in less developed countries, the costs of complying with such environmental standards in addition to the costs of the certification process itself represent a great challenge for producers that want to enter a certification market.

At the local level, initiatives to manage forest ecosystem services can take many forms that depend on several factors, including land ownership, size and governance, forest location and condition, and (of course) the targeted services. Local initiatives can be independent or can derive from initiatives at larger scales, such as the global initiatives discussed above. Other examples of local actions include sustainable forest management and land-use planning, measures to improve forest resilience to disturbance (Fernandes 2013), and actions to restore degraded forests and deforested land, which can help to restore ecosystem services and biodiversity (Chazdon 2008, Rey Benayas et al. 2009).

Despite the scale of implementation, the success of these initiatives to manage forests and their multiple services will depend on the existence of governance structures and legal frameworks that safeguard access to the forest resources and respect the rights of all forest users, thereby avoiding inequity in the access to benefits (FAO 2012). It will also be very important to invest in building capacity to create conditions suitable for the implementation of sustainable forest management programs and to provide local peoples with the knowledge and tools they require to participate in the design and implementation of those programs (FAO 2012).

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